Methane release from wetlands and watercourses in Europe

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Abstract
This study was conducted to estimate annual CH\(_4\) efflux from wetlands and watercourses in Europe and some adjacent areas. Wet ecosystems were divided into seven categories: ombrotrophic mires, minerotrophic mires, freshwater marshes, saltwater marshes, small lakes, large lakes and rivers. The geographical distribution and total area coverage for each of these respective ecosystems were taken from CORINE 2000, Global Land Cover 2000 [JRC, 2003. Harmonisation, mosaicing and production of the Global Land Cover 2000 database (Beta Version). EUR 20849 EN, Joint Research Center, Ispra, Italy] and ESRI 2003 databases. CH\(_4\) release factors were obtained from an extensive overview of published literature. Less than 3% of the study area of 22,560,000 km\(^2\) consisted of wetlands and watercourses. Large lakes (40%), minerotrophic mires (24%) and ombrotrophic mires (20%) covered almost 85% of the total area of wetlands and watercourses. The total CH\(_4\) release from European wetlands and watercourses was estimated to be 5.2 Tg a\(^{-1}\). CH\(_4\) release from minerotrophic mires (48%), large lakes (24%), and ombrotrophic mires (12%) composed most of the total CH\(_4\) efflux. High variation in the rate of CH\(_4\) release within the main ecosystem types, small number of studies in some ecosystems and ecologically inadequate land-cover classification are the main reasons for the uncertainties of the estimate. A better estimation of European CH\(_4\) effluxes from natural sources, now and future, would require: a much more detailed and ecologically relevant mapping of the area of different types of wetlands and watercourses, and long-term measurements of CH\(_4\) fluxes and their controlling environmental factors in poorly studied types of wetlands and watercourses. Finally, the data could be used for dynamic modelling of CH\(_4\) fluxes in the current and changing environmental conditions.

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1. Introduction

Methane (CH\(_4\)) concentration in the atmosphere has increased from about 715 ppb to over 1770 ppb since 1750 due to the anthropogenic activities (IPCC, 2007). Almost 2/3 of the current CH\(_4\) emissions are anthropogenic. The most important anthropogenic sources of CH\(_4\) are rice paddies, ruminants, leakage from coal mining and natural gas transport, biomass burning and waste disposals (Lelieveld et al., 1998; Wuebbles and Hayhoe, 2002). Atmospheric CH\(_4\) is mainly consumed by chemical reactions, mostly with hydroxyl (OH) radicals. The increase in CH\(_4\) emissions decreases OH concentration and thus lengthens the lifetime of CH\(_4\) in the atmosphere (Lelieveld et al., 1993) which further increases the importance of CH\(_4\) as a greenhouse gas.

In addition to anthropogenic sources and chemical sink, geological sources (Etiope, 2008) and natural ecosystems contribute in the cycling of methane. In wetlands, watercourses or other environments with anoxic, highly reductive conditions, the final stage of organic matter...
decomposition will be CH\textsubscript{4} production (methanogenesis). Methane is produced by methanogens that are strictly anaerobic microorganisms of the archaea type. These utilise very simple carbon (C) compounds like acetic acid (CH\textsubscript{3}COOH), carbon dioxide (CO\textsubscript{2}) and methanol (CH\textsubscript{3}OH) for their energy production (Brasseur and Chatfield, 1991; Conrad, 1999). The rate of CH\textsubscript{4} production is reduced when alternative electron acceptors (e.g., NO\textsubscript{3}, SO\textsubscript{4}, Fe(III) and Mn(IV)) are available (Lovley and Phillips, 1987; Oremland, 1988). Part of the CH\textsubscript{4} produced is consumed by methanotrophs, CH\textsubscript{4}-oxidising bacteria (Zehnder and Brock, 1980; Holzapfel-Pschorn et al., 1986). Oxidation can be negligible, but may also reduce all of the CH\textsubscript{4} release depending on the time of the season and the ecosystem type (Rudd and Hamilton, 1978; Moosavi and Crill, 1998; Popp et al., 1999; Bastviken et al., 2002). The difference between CH\textsubscript{4} production and oxidation in the ecosystem determines the net flux of CH\textsubscript{4} between the soil and atmosphere. Dry environments, such as upland taiga and tundra soils, are net sinks of the atmospheric CH\textsubscript{4} (Whalen et al., 1991; Hanson and Hanson, 1996) while water-saturated environments with organic deposits are net sources of CH\textsubscript{4} (Dacey and Klug, 1979; Svensson and Rosswall, 1984). There are numerous studies concerning the rate of CH\textsubscript{4} fluxes in different types of ecosystems but just few estimates about the overall geographical CH\textsubscript{4} fluxes.

This study was conducted to estimate CH\textsubscript{4} release from “natural” wetlands and watercourses in the European continent (see Fig. 1). As hardly any areas in Europe can be regarded in fully natural state, for this study “natural” covers all wetlands and watercourses according to land-cover information, i.e., including artificial reservoirs. Thus, these estimates supplement national greenhouse gas inventories giving updated background information about CH\textsubscript{4} release from natural sources in Europe. Because CH\textsubscript{4} cycling is vulnerable to changes in land-use, atmospheric chemistry and climate, the potential future trends are also discussed. In addition, qualitative uncertainty and semi-quantitative sensitivity analyses were used to point out the highest uncertainties in the estimation, i.e., the needs for future research.

2. Material and methods

2.1. Databases

Natural waterlogged CH\textsubscript{4} sources were divided on the following ecosystem types: ombrotrophic mires, minerotrophic mires, freshwater marshes, saltwater marshes, small lakes, large lakes and rivers. The area estimates for those ecosystems was gathered from different land-cover databases. CORINE 2000 was used to assess wetlands where available, for all other countries Global Land Cover 2000 (JRC). Combination of these databases on a 100 m \times 100 m resolution has been described in detail by Köble et al. (2008)—see Table 1 for classes used. As the databases do not differentiate ombrotrophic and minerotrophic mires, separation takes advantage of the known geographical distribution of main mire complex types (Häyriinen 1970/1978 according to Katz, 1948; Eurola, 1962; Moore and Bellamy, 1974; Botch and Masing, 1983). Mires in mountains outside the area of blanket bogs (ombrotrophic) were regarded minerotrophic in Scandinavia and in proportion of 1:1 ombrotrophic and minerotrophic elsewhere. Correspondingly, mires on the transition zone between marshes (minerotrophic) and raised bogs (ombrotrophic) were considered to represent in equal proportion both ombrotrophic and minerotrophic mires.

Fig. 1. Estimated rate of CH\textsubscript{4} release (in kg CH\textsubscript{4} per grid cell and year) in the NatAir grid.
Table 1
Description of the studied ecosystems and database classes used for the estimation of their area

<table>
<thead>
<tr>
<th>Ecosystem type</th>
<th>Description</th>
<th>Database classes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ombrotrophic mire</td>
<td>A wetland ecosystem receiving nutrients and water almost solely from precipitation.</td>
<td>412 Peatbogs&lt;sup&gt;C&lt;/sup&gt;, 127 Wetland—Bogs and Marshes&lt;sup&gt;C&lt;/sup&gt;</td>
</tr>
<tr>
<td>Minerotrophic mire</td>
<td>A wetland ecosystem receiving water and nutrients mainly from the adjacent mineral soil ecosystems.</td>
<td>412 Peatbogs&lt;sup&gt;C&lt;/sup&gt;, 127 Wetland—Bogs and Marshes&lt;sup&gt;C&lt;/sup&gt;, 128 Wetland—Palsa Bogs&lt;sup&gt;C&lt;/sup&gt;, 129 Wetland—Riparian Vegetation&lt;sup&gt;C&lt;/sup&gt;</td>
</tr>
<tr>
<td>Freshwater marsh</td>
<td>A wetland maintained by the freshwater supply from the surrounding watercourse.</td>
<td>411 Inland Marshes&lt;sup&gt;C&lt;/sup&gt;</td>
</tr>
<tr>
<td>Saltwater marsh</td>
<td>A wetland maintained by the fresh and salt water supply from the surrounding watercourses.</td>
<td>421 Salt-marshes&lt;sup&gt;C&lt;/sup&gt;</td>
</tr>
<tr>
<td>Small lake</td>
<td>A water ecosystem with at least partly standing water. Size $&lt; 1$ km$^2$.</td>
<td>Europe Water Layer&lt;sup&gt;E&lt;/sup&gt;</td>
</tr>
<tr>
<td>Large lake</td>
<td>A water ecosystem with at least partly standing water. Size $&gt; 1$ km$^2$.</td>
<td>Europe Water Layer&lt;sup&gt;E&lt;/sup&gt;</td>
</tr>
<tr>
<td>River</td>
<td>A water ecosystem with flowing water.</td>
<td>Europe Water Layer&lt;sup&gt;E&lt;/sup&gt;</td>
</tr>
</tbody>
</table>


The area of waterbodies (lakes, rivers) was estimated from the ESRI database, Europe Water Layer (ESRI, 2003), providing a spatial resolution of $\sim 25$ m. Smaller water bodies are considered to be negligible. Differentiation between lakes and rivers was done by using aspect ratios of the respective water body, i.e., rivers were identified by their long shorelines with respect to their surface area. Area and circumference of individual water polygons were assessed using GIS functions, and water polygons exhibiting a dimensionless aspect ratio (defined as the area divided by the squared circumference) larger than an empirical factor were assigned lakes, those smaller were considered rivers. Differentiation between small and large lakes was performed on GIS area alone, based on an empirical separation at 1 km$^2$.

2.2. Literature review

Annual methane efflux from the European wetlands and watercourses was estimated by multiplying the area of ecosystem types with the appropriate release factors (g CH$_4$-C m$^{-2}$a$^{-1}$) adopted from literature (Table 2). Referred studies represent different climatic zones concentrating on boreal and temperate zones as the location of wetlands and watercourses in Europe. A variety of methods developed for the monitoring of gas exchange between the biosphere and atmosphere (e.g., closed chamber, eddy covariance, concentration in water) were utilised in the referred studies. Only the release estimates based on repeated measurements in the studied site during the year or growing season were accepted. When the

Table 2
Methane efflux estimates (g CH$_4$-C m$^{-2}$a$^{-1}$) and their range for different wetland and watercourse types

<table>
<thead>
<tr>
<th>Ecosystem type</th>
<th>Best estimate&lt;sup&gt;a&lt;/sup&gt;</th>
<th>Min&lt;sup&gt;a&lt;/sup&gt;</th>
<th>Max&lt;sup&gt;a&lt;/sup&gt;</th>
<th>Number of estimates (European)</th>
<th>Uncertainty&lt;sup&gt;b&lt;/sup&gt;</th>
<th>Sensitivity (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ombrotrophic mire</td>
<td>4</td>
<td>0.07</td>
<td>15</td>
<td>18 (16)</td>
<td>B</td>
<td>47</td>
</tr>
<tr>
<td>Minerotrophic mire</td>
<td>13</td>
<td>0.09</td>
<td>36</td>
<td>45 (36)</td>
<td>B</td>
<td>132</td>
</tr>
<tr>
<td>Freshwater marsh</td>
<td>35</td>
<td>0.1</td>
<td>68</td>
<td>5 (1)</td>
<td>D</td>
<td>15</td>
</tr>
<tr>
<td>Saltwater marsh</td>
<td>3</td>
<td>0</td>
<td>57</td>
<td>5 (2)</td>
<td>D</td>
<td>4</td>
</tr>
<tr>
<td>Small lake (SL)</td>
<td>6</td>
<td>0.1</td>
<td>23</td>
<td>31 (15)</td>
<td>C</td>
<td>17</td>
</tr>
<tr>
<td>Large lake (LL)</td>
<td>4</td>
<td>0.001</td>
<td>8</td>
<td>4 (3)</td>
<td>D</td>
<td>47</td>
</tr>
<tr>
<td>River (R)</td>
<td>2</td>
<td>0.001</td>
<td>10</td>
<td>10 (5)</td>
<td>D</td>
<td>14</td>
</tr>
</tbody>
</table>

<sup>a</sup> Referred publications are marked with the letter codes shown after the name of ecosystem type.


<sup>b</sup> Qualitative uncertainty scale (from UNECE 2004)---A: an estimate based on a large number of measurements made at a large number of facilities that fully represent the sector. Typical error rate is 10–30%; B: an estimate based on a large number of measurements made at a large number of facilities that represent a large part of the sector. Typical error rate is 20–60%; C: an estimate based on a number of measurements made at a small number of representative facilities or an engineering. Typical error rate is 50–150%; D: an estimate based on single measurements, or an engineering calculations derived from a number of relevant facts. Typical error rate is 100–300%; E: an estimate based on a engineering calculations derived from assumptions only. Typical error rate is order of magnitude.
estimates covered the growing season only, 15% of the total annual efflux were assumed to be released during the "dormant" season. The mean value of published estimates was used directly as a release factor in case of rivers, small lakes, large lakes, ombrotrophic mires and minerotrophic mires. For freshwater marshes, half of the marsh area was assumed to represent highly productive conditions with high fluxes (60–90 g CH$_4$ m$^{-2}$ a$^{-1}$, e.g., Phragmites australis growths) and the other half moderate fluxes (10–20 g CH$_4$ m$^{-2}$ a$^{-1}$). In case of saltwater marshes, median was used in order to avoid excessive effect of one oligohaline $R$ australis site (76 g CH$_4$ m$^{-2}$ a$^{-1}$) to overall mean of few values (other four values between 0 and 6 g CH$_4$ m$^{-2}$ a$^{-1}$). This decision was supported by the fact that sulphate in saline ecosystems inhibits methanogenesis (DeLaune et al., 1983; Lu et al., 1999) and thus the realistic CH$_4$ release factor for saltmarshes have to remain lower than that of other ecosystems.

2.3. Uncertainty and sensitivity analyses

Due to the simplified nature of CH$_4$ release calculations, the uncertainty of the final efflux estimates could be evaluated only qualitatively (Table 2). This evaluation was performed using the scale created for atmospheric emissions (UNECE, 2004). Sensitivity analysis was performed quantitatively with the main objective of comparing ecosystems that would contribute more to the total amount of estimated CH$_4$ release. As the uncertainty of the areas in the land-cover classes is small (Mücher, 2000), only CH$_4$ release factors were varied in the sensitivity analysis. This variation was done by using the best estimate and the range of the CH$_4$ efflux for each ecosystem (Table 2). The importance of the factors was judged using results obtained with simple estimations that would tell us the relative change in total CH$_4$ emissions. These calculations followed the method suggested by Leneman et al. (1998). A sensitivity parameter was calculated as

$$\text{Sens} = \left( \frac{E(S_{\text{max}}, A) - E(S_{\text{min}}, A)}{E(S_{0}, A)} \right) \times 100,$$

where Sens is the sensitivity, $E$ the CH$_4$ efflux for all wetland sources calculated with either, $S_{\text{max}}$ the maximum value of the parameter to be evaluated, $A$ the value of all other parameters (fixed at their respective best estimates), $S_{\text{min}}$ the minimum value of the parameter to be evaluated, and $S_0$ is the best estimate of the parameter to be evaluated. In this particular case, $S$ and $A$ variables are release factors and areas, respectively.

3. Results

The study area of 22,560,000 km$^2$ comprises the European continent and some neighbouring land and sea regions (Fig. 1). Less than 3% of that area consisted of wetlands and watercourses, which are geographically concentrated on the northern parts of the study region. Large lakes, minerotrophic mires and ombrotrophic mires covered, respectively, 40%, 24% and 20% of the total area of wetlands and watercourses (Table 3).

Annual CH$_4$ release estimates vary greatly in each ecosystem type (Table 2). The highest effluxes can be found from the most productive ecosystems like freshwater marshes, small lakes and minerotrophic mires. The small number of studies available, especially in Europe, increased the uncertainty of the release factor in case of both marsh types and all watercourse types.

The total CH$_4$ release from European wetlands and watercourses was estimated to be 5.2 Tg a$^{-1}$ (Table 3). Approximately 2/3 is released from wetlands and the rest 1/3 from watercourses. When the ecosystems types are considered separately, CH$_4$ release from minerotrophic mires (48%), large lakes (24%), and ombrotrophic mires (12%) composed most of the total CH$_4$ efflux.

The sensitivity analysis allows attributing the variability of input parameters to the overall results. Total CH$_4$ release estimate for wetlands and watercourses was the most sensitive for the changes in the release factor of minerotrophic mires (Table 2). This is due to the wide range of the observed annual fluxes reported in literature, and the large proportion of minerotrophic mires from the total area of wetlands and watercourses. Even though the range of effluxes is much larger for the saltwater marshes, varying of their release factor contributes little to the total CH$_4$ release estimate due to their small area and hence their small contribution to the total release.

### Table 3

<table>
<thead>
<tr>
<th>Ecosystem type</th>
<th>Area (km$^2$)</th>
<th>% of total area</th>
<th>Methane release (Tg CH$_4$ a$^{-1}$)</th>
<th>% of total release</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wetlands</td>
<td>273,510</td>
<td>46</td>
<td>3.60</td>
<td>69</td>
</tr>
<tr>
<td>Ombrotrophic mire (O)</td>
<td>117,404</td>
<td>20</td>
<td>0.63</td>
<td>12</td>
</tr>
<tr>
<td>Minerotrophic mire (M)</td>
<td>143,298</td>
<td>24</td>
<td>2.49</td>
<td>48</td>
</tr>
<tr>
<td>Freshwater marsh (FM)</td>
<td>10,184</td>
<td>0.48</td>
<td>0.14</td>
<td>3</td>
</tr>
<tr>
<td>Saltwater marsh (SM)</td>
<td>2624</td>
<td>&lt;1</td>
<td>0.01</td>
<td>&lt;1</td>
</tr>
<tr>
<td>Watercourses</td>
<td>315,883</td>
<td>54</td>
<td>1.62</td>
<td>31</td>
</tr>
<tr>
<td>Small lake (SL)</td>
<td>28,212</td>
<td>5</td>
<td>0.23</td>
<td>4</td>
</tr>
<tr>
<td>Large lake (LL)</td>
<td>233,568</td>
<td>40</td>
<td>1.25</td>
<td>24</td>
</tr>
<tr>
<td>River (R)</td>
<td>54,103</td>
<td>9</td>
<td>0.14</td>
<td>3</td>
</tr>
<tr>
<td>Total</td>
<td>589,393</td>
<td>5.22</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Annual CH$_4$ release estimates for wetlands and watercourses are rare. To our knowledge, only Simpson et al. (1999) has estimated CH$_4$ efflux from wetlands in Europe. Our study contains several improvements compared to the earlier estimation. First, Simpson et al. (1999) assumed that CH$_4$ is released only during the "growing season" of 150 days in a year. Several studies have shown, however, that CH$_4$ release during the “dormant” season comprises 5–35% of the annual CH$_4$ release from wetlands and watercourses (Dise, 1992; Alm et al., 1999; Saarnio et al., 2000; Larmola et al., 2004) and this was taken into account in our calculations. Second, the

European-wide CH$_4$ release estimates for wetlands and watercourses are rare. To our knowledge, only Simpson et al. (1999) has estimated CH$_4$ efflux from wetlands in Europe. Our study contains several improvements compared to the earlier estimation. First, Simpson et al. (1999) assumed that CH$_4$ is released only during the "growing season" of 150 days in a year. Several studies have shown, however, that CH$_4$ release during the “dormant” season comprises 5–35% of the annual CH$_4$ release from wetlands and watercourses (Dise, 1992; Alm et al., 1999; Saarnio et al., 2000; Larmola et al., 2004) and this was taken into account in our calculations. Second, the
simplified distribution of climate zones in North America used by Simpson et al. (1999) does not fit to Europe due to the warming effects of Gulf Stream. In the current approach, climate zones were substituted by the natural distribution of wetland types that was considered to take into account effects of climate on annual CH4 efflux. Moreover, the area of wetlands and watercourses covers now in a more complete form the area of Europe, i.e., all countries were accounted for and all watercourse types were accounted for. Finally, CH4 release data was updated with the recent scientific literature. Despite the better spatial and temporal coverage, i.e., bigger area and longer release period, our estimate (5.2 Tg CH4 a\(^{-1}\)) is somewhat lower than that (6.2 Tg CH4 a\(^{-1}\)) of Simpson et al. (1999).

The comparisons for the global estimates can be done only indirectly. Matthews and Fung (1987) have estimated that wetlands between 30° N and 90° N cover 3,149,000 km\(^2\) releasing 75.9 Tg CH4 annually. The coverage of wetlands in our study area represented roughly 9% of the above-mentioned wetland area, but only 5% of its estimated CH4 release. This may indicate that for our study area, our estimate would be lower than that of Matthews and Fung (1987). In contrast, the corresponding comparison for the estimate (20 Tg CH4 a\(^{-1}\)) by Christensen et al. (1996) for the northern wetlands (>50° N, 2,702,000 km\(^2\)) indicate that our estimated release rates (3.2 Tg CH4 a\(^{-1}\) from 256,502 km\(^2\)) would be slightly higher than those in their calculations and very similar (1.1 Tg CH4 a\(^{-1}\) from 82,684 km\(^2\)) than that (19.5 Tg CH4 a\(^{-1}\)) in Crill et al. (1992) for boreal peatlands (45°–60° N, 1,495,000 km\(^2\)). The estimations of both Matthews and Fung (1987) and Crill et al. (1992) are based on the release factors, and “annual” fluxes that have been calculated only for the growing season. Only Christensen et al. (1996) have taken into account winter efflux and their CH4 release estimates are based on the constant proportion (3%) from the heterotrophic respiration modelled for the wetland areas. All the mentioned studies used the areas estimated by Matthews and Fung (1987).

4.2. Uncertainties reveal the future research needs

The uncertainties of the CH4 release estimates arise from uncertainties in the estimation of the area of ecosystem types and from their internal heterogeneity. In principle, the extent of area corresponding to the certain land-cover category is relatively well known. According to the report of Mücher (2000), the area estimates based on remote sensing are within 10% of the true statistical area in general. In the case of this study, the comparison of database areas with the published statistics of wetland areas (Lappalainen, 1996) seemed to agree very well in some countries even while manifold errors remained in some cases. Instead, a more significant problem is that the categories of the land-cover databases do not serve ecological studies. For example, the category called “Peatbogs” in CORINE 2000 database included all types of minerotrophic mires, ombrotrophic mires and peat extraction areas. Ecologically all these main types are very different habitats and differ significantly in their CH4 dynamic. Thus, this lumped category was divided in minerotrophic and ombrotrophic mires according to their general geographical distribution. The proportion of peat extracting areas was assumed to be so small that its effect on the total CH4 efflux would remain behind the uncertainty caused by the natural variation in the rate of CH4 release within the other ecosystem types.

Part of high variation in the annual rate of CH4 release in different kind of ecosystems reflects interannual variability in CH4 efflux. Typically, the highest annual CH4 efflux is clearly less than two-fold compared to the lowest annual efflux on the same site (Saarnio et al., 2007). Instead, the differences in annual CH4 efflux between sites, also within sites belonging to the same ecosystem type, can be multiple. In practise, each of the used seven ecosystem types includes wide variety of subtypes differing in abiotic and biotic factors controlling CH4 production, oxidation and release. Finnish mires alone have been already classified into 30 (Laine and Vasander, 1996) or even over 100 subtypes (Ruhijärvi, 1983) on the basis of the variation in vegetation reflecting local hydrogeochemical conditions. Correspondingly, Finnish lakes alone have been classified on 10 botanical main types (Maristo, 1941). As the published data on annual CH4 effluxes in Europe is scarce for the most of the main ecosystem types, a more detailed subdivision of wetland and watercourse types would currently not be helpful. The lack of information seemed to be highest in case of rivers, large lakes, small lakes, saltwater marshes, freshwater marshes and forest growing wetlands (Table 2). Geographically, there are less studies focusing on areas located in central and southern Europe. This is probably due to the relatively small area coverage of wet ecosystems in those regions (Fig. 1) that discourages to study such ecosystems. In addition, the number of published release factors for different ecosystem subtypes do not necessarily match with the area coverage of those subtypes. Thus, the importance of some subtypes might have been over- or underestimated in calculations when the mean of published values has been used for each main type, a fact that contributes to the result of the sensitivity analysis.

Alternative approach for the fixed release factors would have been a dynamic modelling of CH4 fluxes. Unfortunately, the information in the reviewed publications did not support a more detailed analysis of the dependence of CH4 release on different abiotic and biotic factors. These relationships have been, however, used to develop simple regression models (Christensen et al., 1996; Saarnio et al., 1997; Juutinen et al., 2003b) or more sophisticated process-orientated models (Walter and Heimann, 2000; Kettunen, 2003; Cui et al., 2005) to estimate CH4 efflux from certain wetland ecosystems. Those models, though, do not cover the whole range of the water-logged ecosystems whose rate of CH4 efflux was estimated in this study. In addition, the required input data sets would not have been available over all of Europe.

4.3. CH4 release is sensitive for anthropogenic impacts

Atmospheric CO2 concentration has increased to levels higher than ever during the past 420,000 years or probably even during the past 2.6 million years (IPCC, 2007). The rate of increase has been unprecedented during the past century due to the anthropogenic emissions, and this
increase is not expected to decline in the near future. The raised concentration of CO2 may increase photosynthesis and biomass production especially in plants which—due to their metabolism—are limited by CO2 availability (so-called C3 plants) (Tissue and Oechel, 1987; Dacey et al., 1994; Ojala et al., 2002; Saarnio et al., 2003). Although the litter produced under raised CO2 concentration might be less decomposable (Frederiksen et al., 2001; Ross et al., 2002; van Groenigen et al., 2005), the increased quantity provides more substrates for decomposers. According to the conference review of Norby and Cotrufo (1998), possible small changes in the litter quality are far less important for decomposition than the changes in the litter quantity. Recently assimilated carbon may also be directly allocated to decomposers, including methanogens, via exudation (Megenigal et al., 1999; Aulakh et al., 2001). Final response of the photosynthesis and decomposition in the ecosystem level is of course dependent on the other affecting factors than solely atmospheric CO2 supply. In the case of wetlands and watercourses, laboratory and field studies indicate that CH4 release will increase under raising CO2 concentration (Dacey et al., 1994; Hutchin et al., 1995; Megenigal and Schlesinger, 1997; Saarnio et al., 2000; Liikanen et al., 2003; Silvola et al., 2003; Vann and Megenigal, 2003).

In addition to the direct effects, CO2 may affect CH4 release indirectly, via climate change. The increased concentration of CO2 and other greenhouse gases in the atmosphere strengthen the greenhouse effect, i.e., warm the lower atmosphere (IPCC, 2007). In general, warming enhances the rate of decomposition (Kirschbaum, 1995; Grogan and Chapin, 2000) and thus affects positively CH4 production (Nozhevnikova et al., 1997; Saarnio and Silvola, 1999) as well as CH4 oxidation (Boekx and Van Cleemput, 1996; Saarnio and Silvola, 1999). Few warming experiments of wetland ecosystems indicate changes for the structure and function of vegetation (McKee et al., 2002; Boelmann et al., 2003; Dorrepaal et al., 2003; Weltzin et al., 2003) and predict increase in CH4 release (Oberbauer et al., 1998). On the other hand, the shortening of the ice-covered period might decrease CH4 release from watercourses, as the oxygen-depleted period of the water column would remain shorter (Huttunen et al., 2001).

As hydrology is a major factor affecting C cycling, the indirect consequences of warming, i.e., changes in the regional distribution and timing of the precipitation and increased evapotranspiration are probably the most crucial for the feedback of CH4 release from wetlands and watercourses on climate change. According to the climatic scenarios, precipitation and the frequency of floods will increase in northern Europe whereas in southern Europe precipitation decreases and thus the frequency of droughts increases (Lehner et al., 2006; IPCC, 2007). This kind of changes would support the current pattern, i.e., that wetlands and watercourses will remain mostly concentrated on the northern parts of the continent (Fig. 1). Although wetness favours anaerobic processes, extended flood period may, however, also decrease CH4 release from lakes by shortening the period of plant-mediated CH4 efflux (Juuittinen et al., 2003b). In addition, the possible increase in precipitation in northern regions is projected to happen in winter (Moore et al., 1998; Jylhä et al., 2004) and thus drought periods during the growing season may become more common also in northern latitudes as it has been observed during past decades. Overall, Moore et al. (1998) have estimated that CH4 release will decrease from Canadian mires in the progress of climate change due to the lowered water levels. Nevertheless, they have also predicted that there will be as much variation in response to climate change within a mire as there will be among the peatland regions. The same can be expected in European wetlands and watercourses.

Land-use changes can also have contradictory effects on CH4 release. Drainage of wetlands for agriculture, forestry, peat mining or construction of urban areas decreases CH4 production (Nykänen et al., 1998; Strack et al., 2004). Opposite to that restoration of wetlands and watercourses favours natural CH4 release (Tuittila et al., 2000; Marinier et al., 2004). Pollution may enhance CH4 release, e.g., by via eutrophication of watercourses due to the consequent increase in biomass production (Juuittinen et al., 2003a), or an increase in the atmospheric sulphur (S) or nitrogen (N) deposition may decrease CH4 efflux (Gauci et al., 2002; Silvola et al., 2003). The increasing ozone (O3) concentration in the lower atmosphere may also stress plants causing various structural and functional changes which may affect C cycling in the long-run (Rinnan et al., 2003). As methane release from wetlands and watercourses will encounter several, partly contradictory changes in controlling factors both the direction and magnitude of the change in the rate of CH4 efflux will certainly vary between ecosystems and regions in future.

5. Concluding remarks

When CH4 emissions are estimated, it is always worth to remember that CH4 release from wetlands and watercourses is largely part of the natural C cycling participating in the maintaining of natural greenhouse effect, i.e., it is not responsible for the current climate change. As noted by IPCC (2007), the major contributors to the atmospheric CH4 concentration likely have been identified but their quantification is often affected with high uncertainty due to the difficulty in assessing release rates of highly variable biospheric sources. A better estimation of European CH4 effluxes from natural sources would require: (1) a much more detailed and ecologically relevant mapping of the area of different types of wetlands and watercourses, and (2) long-term measurements of CH4 fluxes and their controlling environmental factors in non-studied types of wetlands and watercourses. In practise, plant communities, which reflect also local environmental conditions, should be utilised more extensively as a basis for land-cover classes. After the matching of current knowledge with the improved land-cover classes, new studies could be concentrated on the largest gaps. An alternative to the release factor approach is a process-based modelling of annual CH4 fluxes, that would in addition of the above-mentioned matters require: (3) the development of new and/or modification of current models to cover all the ecological variation of European wetlands and watercourses, and (4) an extensive observation network for abiotic and biotic factors regulating the CH4 dynamic to
provide an input data for models. Relevant environmental factors concurrently affecting rate of CH4 release include depth of the water level, temperature, density of aerenchymal plant species, rate of primary production in wetlands and concentration of alternative electron acceptors (DeLaune et al., 1983; Saarnio et al., 1997), and in watercourses in addition concentration of total phosphorus (P), dissolved organic carbon (DOC) and CH4, lake size and anoxic volume fraction (Bastviken et al., 2004; Bergström et al., 2007). In future, changes in land-use as well as biospheric and atmospheric pollution will certainly lead to changes in CH4 release from wetlands and watercourses. Understanding the combined effects of different factors will probably require complex modelling based on the field observations and experimental research.

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